Science of the Total Environment 470–471 (2014) 800–817

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Review

Comparison of the Chernobyl and Fukushima nuclear accidents: A review of the environmental impacts

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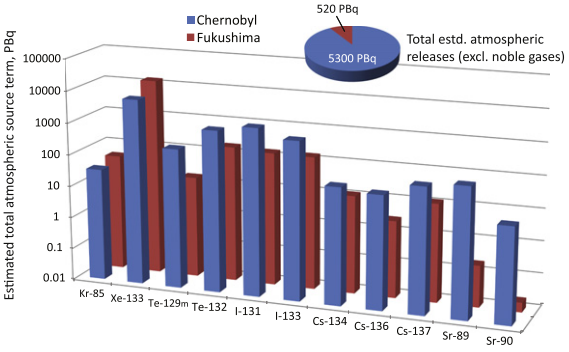
HIGHLIGHTS

- The environmental effects of Chernobyl and Fukushima are compared.
- Releases of radionuclides from Chernobyl exceeded Fukushima by an order of magnitude.
- Chernobyl caused more severe radiation-related health effects.
- Overall, Chernobyl was a much more severe nuclear accident than Fukushima.
- Psychological effects are neglected but important consequences of nuclear accidents.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 25 July 2013
Received in revised form 7 October 2013
Accepted 8 October 2013
Available online 2 November 2013

Keywords:
Fukushima
Chernobyl
Radioactivity
Radioecology
Radionuclide contamination
Socioeconomic effects of radiation

ABSTRACT

The environmental impacts of the nuclear accidents of Chernobyl and Fukushima are compared. In almost every respect, the consequences of the Chernobyl accident clearly exceeded those of the Fukushima accident. In both accidents, most of the radioactivity released was due to volatile radionuclides (noble gases, iodine, cesium, tellurium). However, the amount of refractory elements (including actinides) emitted in the course of the Chernobyl accident was approximately four orders of magnitude higher than during the Fukushima accident. For Chernobyl, a total release of 5300 PBq (excluding noble gases) has been established as the most cited source term. For Fukushima, we estimated a total source term of 520 (340–800) PBq. In the course of the Fukushima accident, the majority of the radionuclides (more than 80%) was transported offshore and deposited in the Pacific Ocean. Monitoring campaigns after both accidents reveal that the environmental impact of the Chernobyl accident was much greater than of the Fukushima accident. Both the highly contaminated areas and the evacuated areas are smaller around Fukushima and the projected health effects in Japan are significantly lower than after the Chernobyl accident. This is mainly due to the fact that food safety campaigns and evacuations worked quickly and efficiently after the Fukushima accident. In contrast to Chernobyl, no fatalities due to acute radiation effects occurred in Fukushima.

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http://dx.doi.org/10.1016/j.scitotenv.2013.10.029

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1. Introduction

The accident at the Fukushima Daiichi nuclear power plant (NPP) was one of the biggest environmental disasters in recent years. In public perception, enhanced by media reports, parallels between the nuclear accidents of Fukushima (Japan, 2011) and Chernobyl (Ukraine, 1986) have often been drawn. Both accidents have been rated on the International Atomic Energy Agency (IAEA) International Nuclear and Radiological Event Scale (INES) as a "Major Accident" as INES 7. However, are the accidents as comparable as suggested by this rating? The nuclear accidents of Chernobyl and Fukushima exhibit some interesting similarities and differences, which warrant comparison. In this review, the main focus is on environmental consequences of both accidents, the causes of the accidents, the types and amounts of radionuclides released, the areas of contamination, the environmental media affected, and a brief discussion of the most relevant radiological aspects including food safety.

2. Causes of the accidents

The Chernobyl nuclear accident happened on 26 April 1986 in the course of a technical test in Unit 4 of the Chernobyl NPP. Inappropriate reactor operation at low power level led to "xenon-poisoning" of the reactor, which was not recognized properly by the reactor staff and caused improper operation of the reactor's control rods (Grishanin, 2010; Smith and Beresford, 2005). This operating error led to thermal destruction of the RBMK-1000 reactor by a sudden power excursion, which ultimately caused at least one (steam) explosion and ignition of the graphite moderators (Michel, 2006). Radionuclides released from the explosion included very short-lived fission products, which resulted in very high dose rates in the adjacent areas. After the initial peak release, further releases of radionuclides occurred over 10 days due to the graphite fire.

On 11 March 2011, the magnitude 9.0 East Japan Earthquake (also referred to as Tohoku Earthquake) occurred at 14:46 (local time) with an epicenter in the Pacific Ocean 130 km east of Sendai (Japan) and 163 km northeast of the Fukushima NPP (Thielen, 2012). The earthquake caused a devastating tsunami that reached heights of up to 40.5 m and caused massive destruction along the coast line. The tsunami rolled as much as 10 km inland (Hamada and Ogino, 2012), causing 15,854 confirmed fatalities and 3089 missing persons (as of 28 March 2012) (Hamada et al., 2012).

The Fukushima Daiichi NPP was operated by the Tokyo Electric Power Company (TEPCO) and consisted of six boiling water reactors with a combined power capability of 5480 MWe (Schwantes et al., 2012). The reactors were brought into operation between 1971 and 1979 and were protected by a 10 m sea wall (Lipscy et al., 2013). The tsunami, however reached as high as 14 m at the plant site. Three of the six reactors (Units 1, 2, and 3) were in operation at the time of the earthquake, but the first seismic signals of the earthquake triggered an automatic shutdown of the reactors. The tsunami reached the site of the NPP at 15:38. It flooded, damaged, and blocked the water intake buildings of the NPP and destroyed the diesel generators, leaving the main cooling systems inoperable due to a complete station blackout. This also included the cooling systems for the spent fuel pools of reactor Units 4, 5, and 6 (Thielen, 2012). Under these circumstances, the battery-driven reactor core isolation pumps remained the only method of cooling for the reactor pressure vessels. The reactor core isolation pump is driven by steam from the pressure vessel, and the steam is discharged into the reactor condensation chamber, while simultaneously pumping water from the condensation chamber into the vessel. However, there was no heat removal from the building via the condensation chamber, and the reactor core isolation pump eventually stopped functioning. After the loss of battery power or pump failure (Unit 1: 11 March; Unit 2: 14 March; Unit 3: 13 March), the reactors were left uncooled (Braun, 2011). At that time, the decay heat of the fission products was still in the range of 20 MW, which caused damage and partial meltdown of the fuel elements. Tanabe (2012) found that core damage started at 1200 K due to ballooning and bursting of the fuel cladding. Core material melting started at 1500 K. Kirchner et al. (2012) estimated that the temperature of the core, however, remained below 2670 K, so that refractory elements were mobilized only to a minor extent. At the temperatures reached, however, the redox-reaction between zirconium and water takes place (the reaction initiates at temperatures >1170 K and becomes autocatalytic >1570 K (Schwantes et al., 2012)), causing the formation of large amounts of hydrogen gas. During venting operations (Blandford and Ahn, 2012) for overpressure relief, both radionuclides and hydrogen gas were released into the service floor level of the reactor buildings, mixing with air. Three massive oxy-hydrogen gas explosions subsequently damaged Unit 1, 3, and 4 buildings. Unit 2 was damaged due to a hydrogen explosion in the condensation chamber.

In contrast to Chernobyl, Fukushima reactors were equipped with a concrete containment building. The explosions at Fukushima were solely of chemical nature (hydrogen explosions) and affected the reactor buildings but, based on the best available information, not the reactor pressure vessels or the reactors themselves. The release characteristics were distinct from the Chernobyl accident. Releases of only gas phase radionuclides occurred in the course of venting operations to relieve over-pressure inside the vessel, after approximately one day delay. In contrast to the uncontrolled, continuous releases of Chernobyl with peak releases in the very beginning, the venting operations at Fukushima NPP happened in pulses over a time span of more than a week, and were often conducted under advantageous weather conditions that transported approximately 80% of the radionuclides offshore (Morino et al., 2011).

3. Types and amounts of released radionuclides

The radionuclide source terms after releases from major nuclear accidents are obtained by model simulations with distinct assumptions and preconditions. This explains the variability of early estimates. For Chernobyl, a value of 5300 PBq (1 PBq = 10^{15} Bq) for the total activity released (excluding noble gases) has been established as the most cited source term in recent literature (UNSCEAR, 2000). Later, the release of refractory elements was adjusted to a 50% lower value. These changes are mostly academic in nature and neither dramatically influence the assessment of radiation doses nor the estimation of the

Table 1
Comparison of the atmospheric release estimates of radionuclides for the nuclear accidents at Chernobyl and Fukushima.

Radionuclide	$T_{1/2}$	Activity (PBq)			
		Chernobyl	Reference	Fukushima[a] (atmospheric releases)	Reference
Noble gases					
^{85}Kr	10.75 y	33	(Dreicer et al., 1996)	44	(Ahlswede et al., 2013)
^{133}Xe	5.25 d	6500	(Dreicer et al., 1996)	**14,000**	(Stohl et al., 2012)
				15,300	(Stohl et al., 2011)
Volatile elements					
^{3}H	12.3 y	1.4 (inventory)	(Kirchner and Noack, 1988)		
^{129m}Te	33.6 d	240	(Dreicer et al., 1996)	~15	This study[c]
^{132}Te	3.20 d	**~1150**	(UNSCEAR, 2008)	**~180**	This study[d]
		1000	(Dreicer et al., 1996)	88	(Tagami et al., 2013)
^{129}I	15.7E6 y	$\mathbf{4 \times 10^{-5} - 4.8 \times 10^{-5}}$	(Aldahan et al., 2007; Kashparov et al., 2003)	$\mathbf{5.5 \times 10^{-5}}$	(Hou et al., 2013)
				6.6×10^{-6}	This study[e]
		8.4×10^{-6}	(Paul et al., 1987)		
^{131}I	8.03 d	**~1760**	(UNSCEAR, 2008)	**150**[f]	(Chino et al., 2011)
		1200–1700	(Dreicer et al., 1996)	130–160	(Hamada and Ogino, 2012)
				190–380	(Winiarek et al., 2012)
				65.2	(Ten Hoeve and Jacobson, 2012)
				200	(Kobayashi et al., 2013)
^{133}I	20.8 h	**910**	(UNSCEAR, 2008)	146	This study[g]
		2500	(Dreicer et al., 1996)		
^{134}Cs	2.07 y	~47	(Dreicer et al., 1996; UNSCEAR, 2008)	**11.8**	This study[h]
				18	(Hamada and Ogino, 2012)
^{136}Cs	13.0 d	36	(Dreicer et al., 1996)	**2.6**	This study[i]
				2.2	This study[j]
^{137}Cs	30.1 d	**85**	(UNSCEAR, 2008)	**12**[f]	(Chino et al., 2011; Winiarek et al., 2012)
		74–85	(Dreicer et al., 1996)		
		98	(Anspaugh et al., 1988)	13	(Kobayashi et al., 2013)
				6.1–15	(Hamada and Ogino, 2012)
				17	(Ten Hoeve and Jacobson, 2012)
				62.5	(Stohl et al., 2011)
Elements with intermediate volatility					
^{89}Sr	50.5 d	**~115**	(UNSCEAR, 2008)	~0.2	This study[k]
		81	(Dreicer et al., 1996)		
^{90}Sr	28.9 y	**~10**	(UNSCEAR, 2008)	~0.02	This study[l]
		4	(Kashparov et al., 2003)		
		8	(Dreicer et al., 1996)		
^{103}Ru	39.2 d	>**168**	(UNSCEAR, 2008)		
		170	(Dreicer et al., 1996)		
^{106}Ru	372 d	>**73**	(UNSCEAR, 2008)		
		30	(Dreicer et al., 1996)		
^{140}Ba	12.8 d	**240**	(UNSCEAR, 2008)		
		170	(Dreicer et al., 1996)		
Refractory elements[b]					
^{95}Zr	64.0 d	**84**	(UNSCEAR, 2008)		
		87	(Kashparov et al., 2003)		
		170	(Dreicer et al., 1996)		
^{99}Mo	66.0 h	>**72**	(UNSCEAR, 2008)		
		210	(Dreicer et al., 1996)		
^{125}Sb	2.76 y	0.23	(Kashparov et al., 2003)		
^{141}Ce	32.5 d	**84**	(UNSCEAR, 2008)		
		200	(Dreicer et al., 1996)		
^{144}Ce	285 d	**~50**	(UNSCEAR, 2008)		
		140	(Dreicer et al., 1996)		
^{154}Eu	8.60 y	0.13	(Kashparov et al., 2003)		
^{239}Np	2.36 d	**400**	(UNSCEAR, 2008)		
		1700	(Dreicer et al., 1996)		
^{238}Pu	87.7 y	**0.015**	(UNSCEAR, 2008)	$2 \times 10^{-6} - 5 \times 10^{-6}$	This study[m]
		0.03	(Dreicer et al., 1996)		
^{239}Pu	24,100 y	0.013	(Kashparov et al., 2003; UNSCEAR, 2008)		
^{240}Pu	6560 y	0.018	(Kashparov et al., 2003; UNSCEAR, 2008)		
$^{239+240}$Pu		0.031	(UNSCEAR, 2008)	$1.0 \times 10^{-6} - 2.4 \times 10^{-6}$	(Zheng et al., 2012)
^{241}Pu	14.3 y	~2.6	(UNSCEAR, 2008)	$1.1 \times 10^{-4} - 2.6 \times 10^{-4}$	(Zheng et al., 2012)
^{242}Pu	3.76E5 y	4×10^{-5}	(UNSCEAR, 2008)		
^{241}Am	433 y	0.0024	(Kashparov et al., 2003)		
^{242}Cm	163 d	~0.4	(UNSCEAR, 2008)		
^{244}Cm	18.1 y	0.0027	(UNSCEAR, 2000)		
Total (excluding noble gases)		**~5300**[f]	(UNSCEAR, 2000)	**~520** (**340–800**)	This study

total activity of the released radionuclides (UNSCEAR, 2008). In any case, no such source term has yet been established for Fukushima. Early estimations, however, ranged from 10% to approximately 15% of the Chernobyl value (Hamada and Ogino, 2012; Ten Hoeve and Jacobson, 2012; Winiarek et al., 2012). Herein we estimate the total activity of radionuclides released to be 520 PBq (340–800 as lower and upper bounds, respectively). This value was obtained by summation of the most likely individual source terms for the most relevant radionuclides (Table 1). For this estimation, in some cases, more recently published values were preferred over preliminary values. In some cases, such as ^{131}I, the most cited value by Chino et al. (2011) was chosen, which is also close to the arithmetic mean of the other published values. For the individual source term of ^{137}Cs, the value obtained by two independent groups (Chino et al., 2011; Winiarek et al., 2012) was deemed most valid (hence printed in bold). Some values in the source term estimate are based on the correlation of a particular radionuclide in environmental samples with a key radionuclide, such as ^{131}I, ^{137}Cs, or others (see also footnotes of Table 1). This correlation provides good estimates of the source term for isotopes of the same element, but introduces larger uncertainties for different elements which have different physical properties (such as volatility) and are prone to chemical differentiation in nature.

It is obvious from Table 1 that the amount of ^{131}I released from Fukushima was less than 10% of the amount released from Chernobyl. Cesium-137, the next most important fission product, was less than 15% of the Chernobyl total. The overall releases from Fukushima Daiichi, according to Table 1, were 10% of the Chernobyl releases. Due to the different release characteristics, the spectrum of discharged radionuclides is distinct for both nuclear accidents. In case of Fukushima, releases of non-volatile radionuclides were small (noble gases, iodine, cesium, and tellurium are considered volatile). Numerous monitoring campaigns studied the distribution of volatile radionuclides in the northern hemisphere (Evangeliou et al., 2013; Masson et al., 2011; Thakur et al., 2013). Examples of semi-volatile or refractory elements would be ^{90}Sr and actinides (Schneider et al., 2013; Steinhauser et al., 2013b; Zheng et al., 2012). Chernobyl, in contrast, released part of its activity inventory directly into the environment in a dynamic that was driven by nuclear processes and a subsequent graphite fire. This explains the significant releases from Chernobyl of semi-volatile or refractory elements (0.4–1.5% of the reactor core inventory) such as radiostrontium (Gaschak et al., 2011; Kashparov et al., 2001; Vakulovsky et al., 1994) and actinides (Agapkina et al., 1995; Kashparov et al., 2003; Mietelski and Was, 1995; Taira et al., 2013) in addition to the volatile radionuclides. Refractory elements from Chernobyl were mainly deposited within a distance of 100 km from the site due to the large particle sizes (Runde et al., 2010). The doses due to ^{90}Sr intake, however, were relatively small (UNSCEAR, 2000). According to an analysis by Kirchner et al. (2012), the nuclear fuel of Fukushima – in contrast to Chernobyl – did not reach temperatures >2700 K that is necessary for the volatilization of refractory elements, such as actinides.

In terms of radionuclide signatures, radiocesium releases from both accidents were clearly distinct. Fukushima-derived radiocesium carried a ^{134}Cs/^{137}Cs activity ratio of 0.98 ± 0.01 Bq/Bq (Merz et al., 2013), whereas radiocesium from Chernobyl had a ^{134}Cs/^{137}Cs signature of 0.5–0.6 Bq/Bq (Arvela et al., 1990; De Cort et al., 1998; Masson et al., 2011). This difference can be explained by the lower degree of enrichment used for the RBMK reactor fuel (approximately 2%). The RBMK fuel does not allow for comparable burning ages, as in case of Fukushima's BWRs (enrichment approximately 3–4%), and will not build up a comparable amount of ^{134}Cs per ^{137}Cs by neutron activation of stable ^{133}Cs.

4. Areas of contamination and evacuation

Chernobyl's "exclusion zone" initially encompassed a 30 km radius (2800 km^2) around the NPP. Approximately 116000 people within the "exclusion zone" were evacuated to less contaminated areas in the months following the accident. Evacuation began 3–11 days after the accident, which was already late for parts of the affected population (Pröhl et al., 2002). Later, the exclusion zone was extended and covered 4300 km^2 in 1996, in order to contain the areas with the highest radiation levels (IAEA, 1996).

In contrast, more than 80% of the atmospherically-released radionuclides are estimated to have gone offshore from Fukushima, followed by deposition in the Pacific Ocean (Kawamura et al., 2011; Morino et al., 2011). Radionuclides from Fukushima were found in seawater and marine organisms throughout the Pacific (Buesseler et al., 2011; Buesseler et al., 2012; Hou et al., 2013; Madigan et al., 2012; Madigan et al., 2013; Manley and Lowe, 2012; Povinec et al., 2012; Shimura et al., 2012). However, the radiological consequences in countries other than Japan appear negligible, as dilution in the Ocean was found to occur quickly (Behrens et al., 2012). Chernobyl, in contrast, is located in the center of the European continent resulting in significant contamination also outside the evacuation zone (Thomassin et al., 2011), and in many European countries. This includes the Ukraine's neighboring countries Belarus (which is generally regarded as the most affected country of the Chernobyl accident) (Krinitsyn and Pazukhin, 1995; Zhuchenko et al., 2002), the Russian Federation (Andersson and Roed, 2006; Bernhardsson et al., 2011; Korobova et al., 1998), Scandinavia (Holm et al., 1992; Jaworowski et al., 1997; Leppänen et al., 2011), Southern Germany (Bunzl et al., 1995; Daraoui et al., 2012; Schimmack et al., 1997), Austria (Bossew et al., 1995), Greece (Petropoulos et al., 1996), and others (Fig. 1 and Table 2) (De Cort et al., 1998).

The current Fukushima exclusion zone encompasses an area of less than 600 km^2 (Fig. 2.) (Yoshida and Takahashi, 2012). Radionuclides were mainly deposited northwest of the reactor, causing the greatest contamination in a strip approximately 40 km in length (Hirose, 2012; Kinoshita et al., 2011; Yasunari et al., 2011). Despite the fact that the tsunami had destroyed large parts of the coastal infrastructure,

Notes to Table 1:

[a] Comment: If necessary, activities of very short-lived radionuclides were decay corrected to 12 March 2011 (12:00 noon), which is the date of the first releases of radionuclides.

[b] Chernobyl: releases based on the respective radionuclide inventory in Unit 4 and a release of ~1.5% of the fuel in particulate form, see Kashparov et al. (2003).

[c] Estimated from the ^{137}Cs source term from Chino et al. (2011) and the measured ^{129m}Te/^{137}Cs activity ratio of 1.3 from Endo et al. (2012) (disregarding some obvious outliers). The proposed ^{129m}Te/^{137}Cs activity ratio of 4.0 by Tagami et al. (2011) has been found to be inconsistent with the other radiotellurium/radiocesium activity ratios tested in this study. This may be due to chemical fractionation between Cs and Te in the environment.

[d] Estimated from the ^{137}Cs source term from Chino et al. (2011) and the measured ^{132}Te/^{137}Cs activity ratio of 15 from Endo et al. (2012) (disregarding some obvious outliers).

[e] Based on the ^{131}I data from Chino et al. (2011) and the measured atomic ratio for ^{129}I/^{131}I = 31.6 from Miyake et al. (2012).

[f] Most cited value in literature as of May 2013.

[g] Based on the ^{131}I data from Chino et al. (2011) and a measured ^{133}I/^{131}I activity ratio of 0.97 from Amano et al. (2012).

[h] Based on the ^{137}Cs data from Chino et al. (2011) and a measured ^{134}Cs/^{137}Cs activity ratio of 0.98 from Merz et al. (2013).

[i] Based on the ^{137}Cs data from Chino et al. (2011) and a measured ^{136}Cs/^{137}Cs ratio of 0.22 from Tagami et al. (2011) as well as Steinhauser et al. (2013a).

[j] Based on the ^{137}Cs data from Chino et al. (2011) and a measured ^{136}Cs/^{137}Cs ratio of 0.18 from Amano et al. (2012).

[k] Based on the estimation for ^{90}Sr in this study and an initial ^{89}Sr/^{90}Sr activity ratio of 11.8 from Povinec et al. (2012); Casacuberta et al. (2013); Nishihara et al. (2012).

[l] Estimated from the ^{137}Cs source term from Chino et al. (2011) and the ^{90}Sr-^{137}Cs correlation from Steinhauser et al. (2013b) (disregarding one outlier).

[m] Based on the $^{239+240}$Pu data from Zheng et al. (2012) and a predicted ^{238}Pu/$^{239+240}$Pu activity ratio of 1.92 from Schwantes et al. (2012).

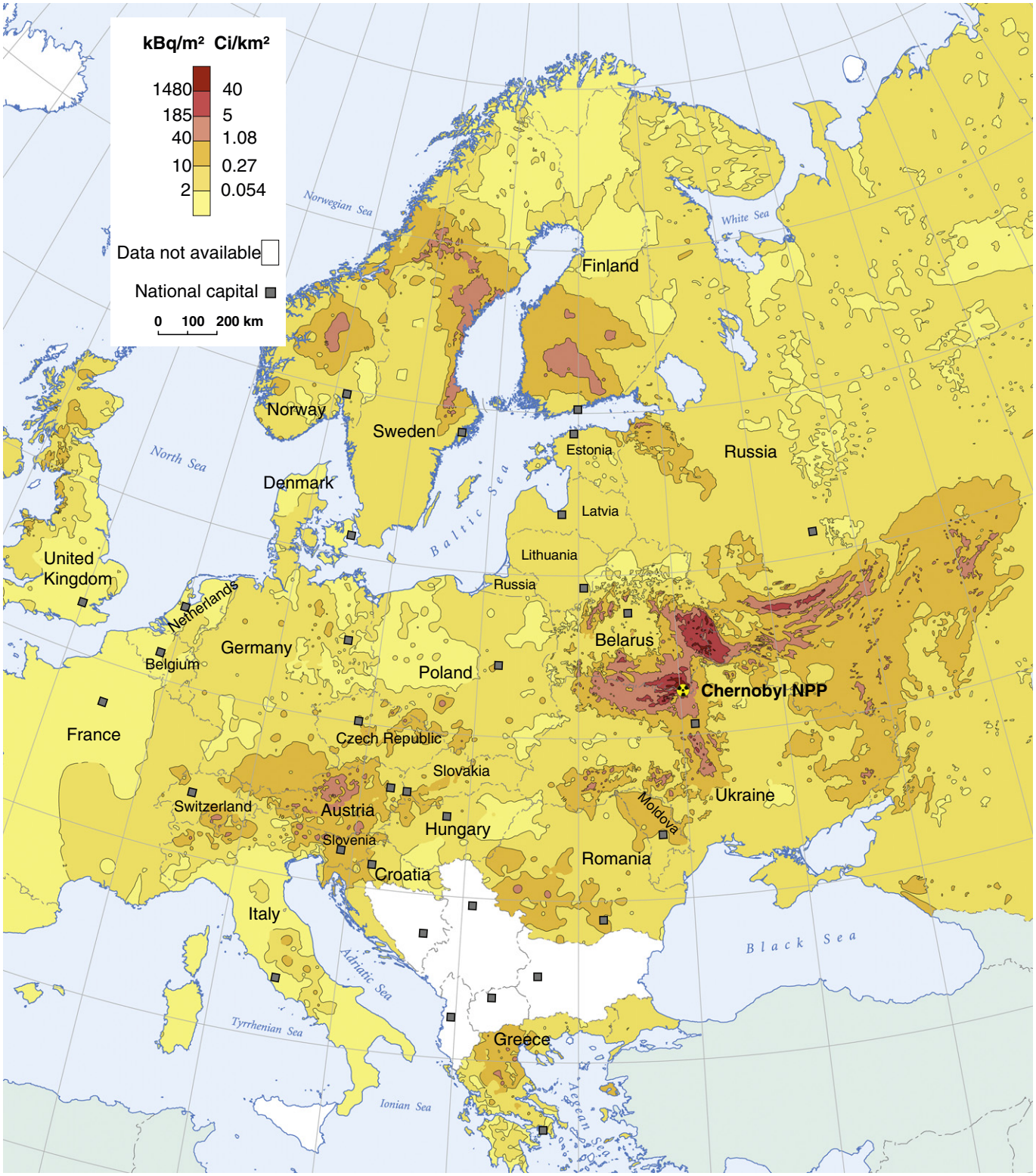

Fig. 1. Surface contamination with ^{137}Cs in Europe after the Chernobyl nuclear accident. Modified after De Cort et al. (1998) Reprinted with permission from the European Commission Joint Research Centre. © EC/IGCE, Roshydromet (Russia)/Minchernobyl (Ukraine)/Belhydromet (Belarus), 1998.

Japanese authorities managed the evacuation of the population from the potentially affected areas remarkably well and fast. The Japanese Nuclear Emergency Response Headquarters (NERHQ) defined the evacuation zones. On 12 March 2011, a 20 km radius around the Fukushima NPP was designated as the "stay-away evacuation" zone. On 15 March 2011, the time of major releases of radionuclides, the surrounding area between 20 and 30 km was declared an "indoor evacuation" zone. On 16 March 2011, NERHQ instructed evacuees

Table 2
Contaminated areas in European countries. Please note a contribution of 2–4 kBq/m^2 due to previous atmospheric nuclear explosions. Taken from Diehl (2003); United Nations (2000).

Country	Areas of the contamination zones (km^2)			
	Zone 4 (37–185 kBq/m^2)	Zone 3 (185–555 kBq/m^2)	Zone 2 (555–1480 kBq/m^2)	Zone 1 (>1480 kBq/m^2)
Russian Federation	49,800	5700	2100	300
Belarus	29,900	10,200	4200	2200
Ukraine	37,200	3200	900	600
Sweden	12,000			
Finland	11,500			
Austria	8600			
Norway	5200			
Bulgaria	4800			
Switzerland	1300			
Greece	1200			
Slovenia	300			
Italy	300			
Moldova	80			

less than 40 years of age to leave the stay-away evacuation zone and to take pills or syrup of stable iodine (Hamada et al., 2012; NERHQ, 2011).

Stable iodine prophylaxis can prevent uptake of radioiodine into the thyroid gland and thus reduce the risk of thyroid cancer. Stable iodine (1.51 million pills for 750,000 people and 6.1 kg powder for 120,000–180,000 people) was distributed in Fukushima Prefecture on 16 March 2011, but only a small number of evacuees actually ingested the stable iodine as the evacuation had already been completed

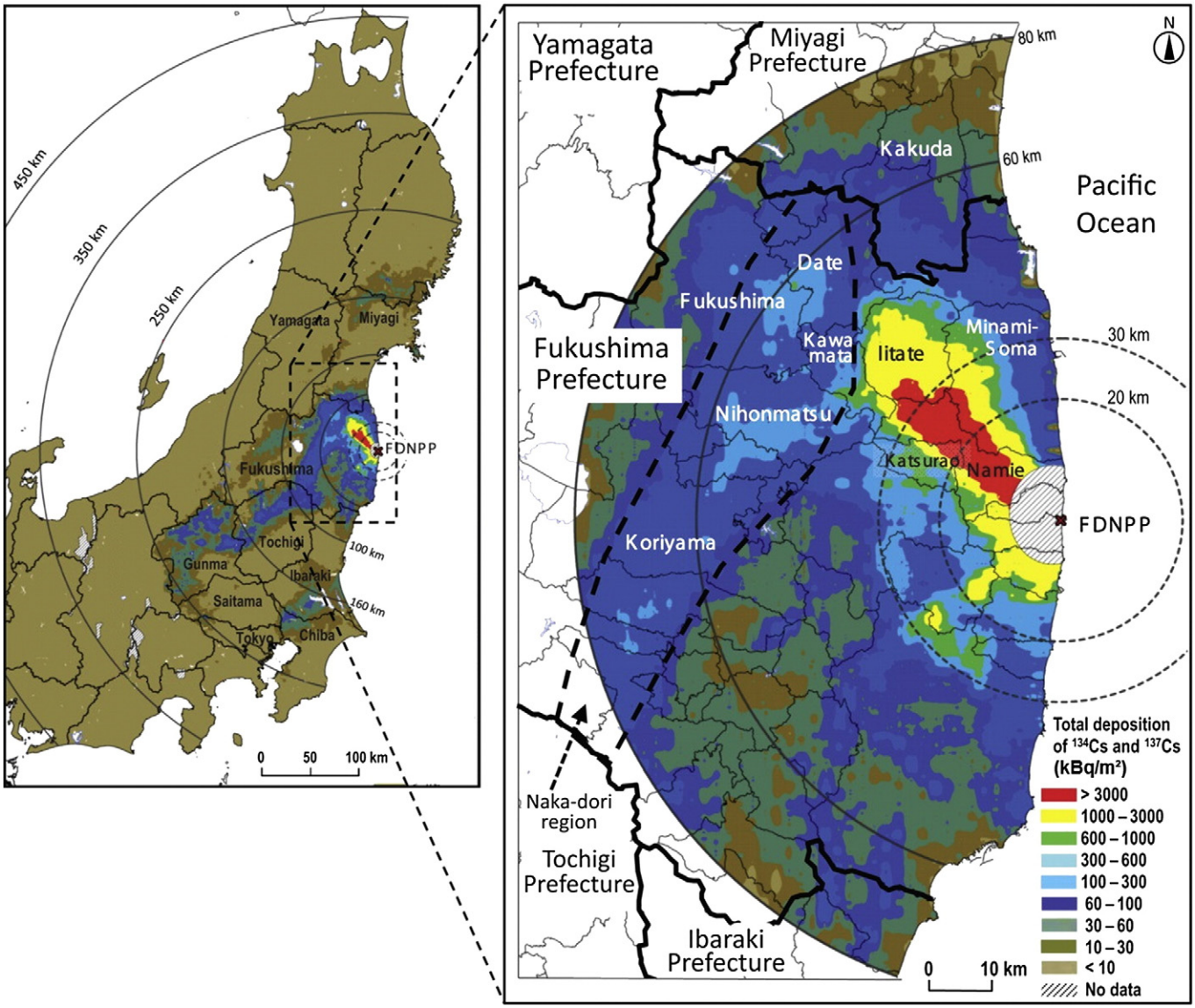

Fig 2. Estimated total deposition of radiocesium after the Fukushima nuclear accident, approximately half of which is ^{137}Cs. Taken from (Yoshida and Takahashi, 2012) and slightly modified. Reprinted with permission from the Mineralogical Society of America © 2012.

previously (Hamada et al., 2012). At this point, the importance and the effectiveness of the rapid evacuation have to be emphasized (Thomassin et al., 2011).

Although the distribution of stable iodine has been reported for Chernobyl in early publications (International Atomic Energy Agency, 1989; International Atomic Energy Agency, 1991), the fact itself or at least the effectiveness of this countermeasure for the region's population was challenged at a later point (Shakhtarin et al., 2003). If this countermeasure was deployed at all, it was probably too late for many inhabitants.

On 22 April 2011, Fukushima's 'indoor evacuation zone' (sheltering zone) was revised to the 'evacuation prepared' zone, where residents should be ready for evacuation or sheltering in case of a deterioration of the situation. However, this excluded the area beyond the 20-km radius where doses >20 mSv for the general public during the course of the next year could not be excluded. This area designated as the 'deliberate evacuation area' was located in northwesterly direction of the NPP (Hamada et al., 2012).

The areas of the individual contamination zones in various countries after the Chernobyl nuclear accident are summarized in Table 2. According to this table, the area around Chernobyl exceeding a level >185 kBq/m^2 encompasses 29,400 km^2. The size of the contamination zone in Japan with levels >185 kBq/m^2 after the Fukushima accident, in comparison, is defined by an area of approximately 1700 km^2 (Ohta, 2011). In Japan, more than 75% of the area is forested, <10% rice paddy fields, <10% other agricultural areas and <5% urban areas. The contaminated areas (>185 kBq/m^2) in Belarus are 43% agricultural areas, 39% forested and 2% rivers and lakes (Ohta, 2011).

5. Environmental impact of the released radionuclides

Both the Chernobyl and Fukushima accidents caused radionuclide contamination of the atmosphere, hydrosphere, biosphere and pedosphere over the entire northern hemisphere. Activity concentrations of the important radionuclides ^{131}I, ^{137}Cs, ^{90}Sr and the sum activity of ^{239}Pu and ^{240}Pu (if available) in air, rainwater, plant material/animal thyroids, and soil are compared in Tables 3a–3d for locations in various distances from the respective NPP (Table 3a: air; Table 3b: rainwater; Table 3c: biosphere; Table 3d: soil). Wherever literature data are available, we compared data from locations at equal or at least comparable distance to the respective NPPs in one row of the tables. Where more than one data are available, the Tables show the maximum values.

Please note that in Table 3d (activity concentrations in soil) long-lived radionuclides such as ^{137}Cs, ^{90}Sr and $^{239+240}$Pu can be present from previous accidents or nuclear weapon tests and exhibit a significant background concentration that is sometimes difficult to distinguish from "fresh" contamination. In this case, radionuclides such as ^{134}Cs or ^{89}Sr can be used as indicators of a recent contamination. The background in the other environmental media (e.g. caused by resuspension of soil particles or uptake of radionuclides by plants) can mostly be regarded as of minor importance.

Table 3a
Selected maximum radionuclide concentrations in air after the accidents of Chernobyl and Fukushima, arranged by distance from the NPPs (activities in Bq·m^{-3} at the time of sampling).

Chernobyl						Fukushima					
Location	Distance to NPP (km)	^{131}I *	^{137}Cs	^{90}Sr	$^{239+240}$Pu	Location	Distance to NPP (km)	^{131}I *	^{137}Cs	^{90}Sr	$^{239+240}$Pu
Chernobyl NPP	0	750,000[a]	120,000[a]	–	–	Fukushima Daiichi	0	5600[s]	–	–	–
Chernobyl, UA	1	58,000[b]	–	–	–	Fukushima Daiichi	10	2100[t]	–	–	–
						Futoba, JP	25	530[u]	6.6[u]	–	–
						Tsukuba, JP	190	32[v]	0.016[v]	–	–
						Takasaki, JP	220	14.7[w]	–	–	–
Berezinsky Natural Res., BY	400	200[c]	9.9[c]	–	–	Wako, JP	220	47[x]	–	–	–
Vilnius, LT	500	45.2[d]	27.9[d]	–	–						
Mikolajki, PL	650	8[e]	1.88[e]	–	–						
Prague, CZ	1100	–	–	–	0.000028[f]	Busan, ROK	1100	0.00132[y)]	0.00125[y]	–	–
Nurmijärvi, FIN	1100	–	10[g]	–	0.000032[g]	Andong, ROK	1100	0.00069[y]	0.00015[y]	–	–
Sweden	1150	12[h]	5[h]	–	–	Daegu, ROK	1120	0.00112[y]	0.00027[y]	–	–
Austria	1200	–	10[i]	0.17[i]	0.00009[i]	Seoul, ROK	1200	0.00129[y]	0.00017[y]	–	–
Thessaloniki, GR	1300	5[j]	2[j]	–	–	Gunsan, ROK	1300	0.00312[y]	0.00045[y]	–	–
						Jeju, ROK	1400	0.00089[y]	0.0005[y]	–	–
						Jilin, CN	1500	0.008[z]	0.00023[z]	–	–
Bologna, IT	1700	6.44[k]	1.9[k]	–	–	Liaoning, CN	1600	0.00147[z]	0.00017[z]	–	–
Monaco	1900	4.63[a]	1.56[a]	–	–						
Paris	2000	2[l]	2[l]	–	–						
Yavne, IL	2200	4.2[m]	–	–	–	Anhui, CN	2200	0.00159[z]	0.00017[z]	–	–
Chilton, UK	2200	0.96[n]	0.5[n]	–	8.5E-7[n]	Taiwan	2400	0.00059[aa]	0.0006[aa]	–	–
						Saipan	2500	0.00999[bb]	0.00148[bb]	–	–
						Manila, RP	3200	0.00048[cc]	0.00012[cc]	–	–
						Wake Island	3200	0.0023[dd]	0.00035[dd]	–	–
						Dalat, VN	3800	0.00019[ee]	0.000036[ee]	–	–
						Midway Island	4000	0.0093[ff]	0.001[ff]	–	–
						Dutch Harbor, AK, US	4350	0.0255[gg]	0.00325[gg]	–	–
Mumbai, IN	5100	–	1.65[o]	–	–	Tibet	4700	0.00032[z]	–	–	–
						Nome, AK, US	4600	0.00159[hh]	0.0044[hh]	–	–
						Oahu, US	6000	0.0244[ii]	0.00444[ii]	–	–
						Ketchikan, AK, US	6500	0.00213[jj]	0.00023[jj]	–	–
New York, US	7500	0.02[p]	0.01[p]	–	–	Dubna, RUS	7400	0.00386[kk]	–	–	–
						Seattle, WA, US	7500	0.0044[ll]	0.00022[ll]	–	–
Chiba, JP	8200	1[q]	0.074[q]	–	–	Vilnius, LT	8000	0.0038[mm]	0.00107[mm]	–	4.45E-8[nn]
						Reykjavik, IS	8600	0.003[oo]	0.00087[oo]	–	–
						Krakow, PL	8700	0.0057[pp]	0.00044[pp]	–	–
Livermore,CA,US	9800	0.074[r]	0.037[r]	–	–	Bremen, D	8900	0.002[qq]	–	–	–
						Tessaloniki, GR	9300	0.000497[rr]	0.000145[rr]	–	–

Tables 3a–3d illustrate that in most cases the monitoring campaigns reported much higher activity concentrations after Chernobyl, when compared with Fukushima. It is important to note that some (soil and plant) analyses after Chernobyl were performed several years after the accident, reducing the apparent activity concentrations due to physical decay and migration into the soil. Naturally, the degree of contamination not only depends on distance but also on weather conditions such as wind direction and precipitation, which, especially for the analysis of the Chernobyl accident remains difficult to determine retrospectively. Local meteorological conditions cause great variability in activity concentrations in environmental media and can result in locations further away being higher in concentration than closer locations. For the same reason, occasionally, activity concentrations in environmental media are found to be higher after the Fukushima accident than after Chernobyl. For example, ^{131}I activity concentrations in rainwater in 2011 collected at San Francisco, USA (16 Bq·L^{-1}; 8100 km from Japan) were approximately 10 times higher after the Fukushima accident (Table 3b), than rainwater sampled in Taiwan (7600 km from Chernobyl) after the Chernobyl accident (1.4 Bq·L^{-1}). However, Tables 3a, 3b, 3c, 3d show a clear picture: in almost every environmental medium, and at almost every distance from the NPPs, the Chernobyl accident caused a much greater degree of contamination, often several orders of magnitude higher than as observed after the Fukushima accident. For example, this is also illustrated by the ^{131}I concentrations in rainwater collected in Japan after the Chernobyl accident (98 Bq·L^{-1}), which was 6 times higher than the above-mentioned rainwater from San Francisco in 2011.

The most obvious difference between both accidents is the presence or absence of refractory elements such as plutonium or semi-volatile ^{90}Sr in the environmental media listed in Tables 3a–3d. Chernobyl emitted at least 0.4% of its fuel inventory (Kashparov et al., 2003). Many reports on semi-volatile ^{103,106}Ru and refractory radionuclides (^{95}Zr, ^{140}La, ^{141,144}Ce, etc.) in air and other media have been published after Chernobyl, even at remote monitoring stations (but could not be included in Tables 3a–3d) (Erlandsson et al., 1987; Irlweck et al., 1993; Ishida et al., 1988; Larsen et al., 1989; Retalis and Pitta, 1989; Whitehead et al., 1988), but hardly any after Fukushima (Shozugawa et al., 2012). Even reports on radionuclides of semi-volatile elements such as strontium from Fukushima are still rare (Povinec et al., 2012; Steinhauser et al., 2013b) — in stark contrast to Chernobyl (Kashparov et al., 2001; Kashparov et al., 2003). Findings of refractory radionuclides in air after 1986 can be explained by the distinct accident dynamics and the fact that fuel debris was emitted into the atmosphere in the course of the Chernobyl accident. More than 90% of the radiostrontium and radioruthenium activities at Chernobyl were emitted in fuel particle form (Kashparov et al., 2003; Kuriny et al., 1993). In contrast, at Fukushima gas phase emissions clearly dominated the releases.

Releases from the Chernobyl accident also included those radionuclides with very short half-lives in the range of seconds and minutes that are characterized by enormous specific activities. This led to the massive radiation damage in the so-called “Red Forest”, where pine trees succumbed due to radiation-induced destruction of plant tissue in the early aftermath of the accident. The name

Notes to Table 3a:
*Particulate fraction only.
– … not detected or not reported
[a]26 April 1986 (Chernobyl, UA); 4–5 May 1986 (Monaco) (Whitehead et al., 1988).
[b]9 May 1986 (Feely et al., 1988; USSRSCUAE (USSR State Committee on the Utilization of Atomic Energy), 1986).
[c](Thakur et al., 2013).
[d]Early May 1986 (Lujaniene et al., 2012b).
[e]30 April 1986 (Jaworowski and Kownacka, 1988).
[f]29 April–5 May 1986 (Holgye, 2008).
[g]28 April 1986 (Paatero et al., 2010).
[h]28–29 April 1986 (Devell et al., 1986).
[i]Late April–5 May 1986 (Irlweck et al., 1993; Irlweck and Wicke, 1998).
[j]5–6 May 1986 (Papastefanou et al., 1988).
[k]1 May 1986 (Gattavecchia et al., 1989).
[l]1 May 1986 (Thomas and Martin, 1986).
[m]5 May 1986 (Paul et al., 1987).
[n]2–3 May 1986 (Cambray et al., 1987).
[o]13 May 1986 (Mishra, 1990).
[p]10–11 May 1986 (Larsen et al., 1989).
[q]4 May 1986 (Higuchi et al., 1988).
[r]12 May 1986 (Beiriger et al., 1988).
[s]21 March 2011, MEXT (Thakur et al., 2013).
[t]20 March 2011, TEPCO (Thakur et al., 2013).
[u]MEXT (Thakur et al., 2013).
[v]16 March 2011 (^{131}I), 1–3 April (^{137}Cs), KEK (Thakur et al., 2013).
[w]15–16 March 2011, CTBTO (Thakur et al., 2013).
[x]22–23 March 2011 (Thakur et al., 2013).
[y]6–7 April 2011 (Kim et al., 2012).
[z]4–9 April 2011, MEP (Thakur et al., 2012).
[aa]6 April 2011 (Huh et al., 2012).
[bb]21 March 2011, US EPA (Thakur et al., 2013).
[cc]24 March 2011, CTBTO (Thakur et al., 2013).
[dd]25 March 2011, CTBTO (Thakur et al., 2013).
[ee]10 April 2011 (Long et al., 2012).
[ff]22–23 March 2011, CTBTO (Thakur et al., 2013).
[gg]19–22 March 2011, US EPA (Thakur et al., 2012).
[hh]23–24 March 2011, US EPA (Thakur et al., 2012).
[ii]21 March 2011, US EPA (Thakur et al., 2013).
[jj]25–26 March 2011 (Miley et al., 2013).
[kk]4 April 2011 (Masson et al., 2011).
[ll]19–24 March 2011 (Diaz Leon et al., 2011).
[mm]28 March–1 April 2011 (Lujaniene et al., 2012b).
[nn]April 2011 (Lujaniene et al., 2012a).
[oo]2 April 2011, IRSA (Thakur et al., 2012).
[pp]29–30 April 2011 (Masson et al., 2011).
[qq]29 March 2011 (Pittauerová et al., 2011).
[rr]4–5 April 2011 (Manolopoulou et al., 2011).

"Red Forest" comes from the ginger-brown color of the pine-needles of the perished trees. Due to the much higher levels of contamination and releases of fuel particles in the vicinity of the Chernobyl NPP, radiation-induced mutations of plants (winter wheat) have been reported (Morgun et al., 1996). The releases from Fukushima, in contrast, began on 12 March 2011, many hours after the emergency shut-down of the reactors following the first seismic signals of the earthquake. This delay of several hours before any appreciable releases was sufficient for the very short-lived fraction of radionuclides to decay, which avoided a comparable ecological devastation like in Chernobyl.

Table 3a offers some interesting insight in the uptake mechanisms of a radionuclide, in particular radioiodine in the thyroids of mammals. Iodine-131 concentrations in human thyroids from Munich (Germany) (Kutschera et al., 1988) were two or three orders of magnitude lower than activity concentrations in the thyroids of grazing animals from closely located regions (Austria and Ulm/Germany) (Tataruch et al., 1988; Van Middlesworth and Loos, 1988). It can be assumed that the activity concentrations in air were roughly comparable in all three areas. This shows that inhalation of radioiodine is only a minor pathway into mammals thyroids compared with the ingestion of contaminated food/pasture. It can be assumed that grass exhibited a much higher contamination level in early May 1986 than human food, which typically consists partly of processed and imported foodstuffs that are not susceptible to local atmospheric ^{131}I pollution. This example further emphasizes the importance of food safety after nuclear accidents.

Apart from radionuclide concentration in air, electrical conductivity can provide much information about a current contamination via the ionization state of air. However, it was shown that if air conductivity was to be used for radioactivity monitoring, a simultaneous aerosol particle size distribution measurement was necessary (Paatero et al., 2010). In any case, airborne radionuclides from the Chernobyl accident significantly increased the electrical conductivity in air in Europe. In Sweden, conductivity increased from 20 fS m^{-1} to 220 fs m^{-1} (Israelsson and Knudsen, 1986; Paatero et al., 2010). A similar effect was observed in Athens (Greece) (Retalis and Pitta, 1989). The station at Helsinki-Vantaa airport (Finland) reported a tenfold increase in conductivity in April–May 1986. The conductivity meter even went over the scale maximum, and a conductivity of 150–200 fS m^{-1} could only be estimated (Paatero et al., 2010; Tuomi, 1989). The normal background level in Finland was reached by the end of summer 1986. In the highly contaminated area around the Chernobyl NPP, in contrast, air conductivity was reported 240–570 times higher than the background even one year after the accident (Yablokov et al., 2009). As far as scientific literature reports on the Fukushima accident are concerned, no comparable conductivity anomalies have been observed after the Fukushima accident.

6. Doses, health effects and projected mortality

More than two decades after the Chernobyl accident, the radiological consequences of the radionuclide releases are well understood. An ultimate and holistic assessment of the health effects, for example in terms of fatalities attributable to the accident, will require further investigations, but perhaps will never be possible (Jacob et al., 2006).

Tokonami et al. (2012) investigated 62 residents and evacuees from heavily contaminated areas around the Fukushima NPP to determine their thyroid doses (inhalation) and found a median committed equivalent dose of 4.2 mSv and 3.5 mSv for children and adults, respectively. The maximum thyroid doses were 23 mSv (children) and 33 mSv (adults). For the mean committed thyroid dose for evacuees from Chernobyl, Tokonami et al. cite a value of 490 mSv (UNSCEAR, 2008). The maximum thyroid doses after Chernobyl ranged as high as 50 Gy (The Chernobyl Forum, 2006). Whereas the main pathway of radioiodine incorporation in Chernobyl was uptake of contaminated milk (UNSCEAR, 2008), exposure through ingestion is considered to contribute negligibly to the internal exposure after the Fukushima nuclear accident.

The thyroid dose after the Chernobyl accident has been associated with an increase in thyroid cancer incidence among those who were children and adolescents at the time of the accident (Kazakov et al., 1992). Also, fetuses whose mothers were exposed to radioiodine received significant thyroid doses, up to 3.2 Gy, with an arithmetic mean of 72 mGy (Likhtarov et al., 2011). As reported by UNSCEAR in 2008, the thyroid cancer incidence in the affected population exceeded 6848 cases for patients who were younger than 18 years in 1986 (5127 of them were under the age of 14 in 1986) (UNSCEAR, 2008). Recent studies suggest that this number may be still increasing (Ivanov et al., 2012). Treatment of thyroid carcinoma often proves to be successful; however, 15 fatalities have been reported as of 2006 (The Chernobyl Forum, 2006). No risk of radiation-related thyroid cancer was found in a group of those exposed as adults (Dickman et al., 2003; Ivanov et al., 2012; Tronko et al., 2006). It is far too early to predict any thyroid cancer incidence for Fukushima, however, first estimates carefully suggest that "the doses to a vast majority of the population in Fukushima were not high enough to expect to see any increase in incidence of cancer and health effects in the future" (Yamashita and Suzuki, 2013). Yamashita and Suzuki (2013), however admit that public concerns about the long-term health effects of the disaster have increased in Japan. Recent news reports indicated 18 cases of thyroid cancer among children in Japan (NHK, 2013). However, such reports must be treated carefully because of the latency period of radiation-induced thyroid cancer. The International Commission on Radiological Protection (ICRP, 2007) indicated a 4-year latency period after exposure. Similarly, a minimum latency of three years was shown after Chernobyl, followed by a linear increase with time after exposure (Heidenreich et al., 1999). In any case, an excess cancer risk associated with childhood exposure was shown to persist for more than 50 years after exposure for the Japanese atomic bomb survivors (Furukawa et al., 2013), which makes long-term monitoring of the affected population indispensable in both the Chernobyl and the Fukushima areas.

Various radiation-induced cancer mortality projections for Chernobyl "liquidators", evacuees, and people living in areas with contamination levels >37 kBq/m^2 have been published in the literature. However, these projections, ranging from a few thousand cancer mortalities (Michel and Voigt, 2006) to numbers as high as 10,000 to 40,000 (Anspaugh et al., 1988; Cardis et al., 2006; NRC, 2006; Ten Hoeve and Jacobson, 2012) that are often based on generic risk estimates. Such risk projections are generally discouraged by the International Commission on Radiological Protection (ICRP) (ICRP, 2007), in particular where they are derived from collective effective doses from trivial individual doses. More importantly, however, statistically significant increases in the number of leukemia cases, solid tumors, birth defects, or genetic defects have not yet been observed for the potentially affected population around Chernobyl (Michel and Voigt, 2006; The Chernobyl Forum, 2006). Early indications of an increased risk of leukemia among 61,000 "liquidators" (The Chernobyl Forum, 2006), seem to have been confirmed in more recent epidemiological studies, together with increased risks of hematological malignancies and of cataracts (Cardis and Hatch, 2011).

In a recent publication, similar risk projections have been employed to estimate the worldwide health effects of the Fukushima nuclear accident (Ten Hoeve and Jacobson, 2012). According to their modeling, radiation exposure is projected to result in an additional 130 (15–1100) cancer-related mortalities and 180 (24–1800) cancer-related morbidities (uncertainties associated with the exposure-dose and dose–response models) worldwide. A reply (Beyea et al., 2013) as well as a previous estimation (Von Hippel, 2011) suggest higher numbers of fatal cancer of around 1000 cases (which is still within the uncertainty given in by Ten Hoeve and Jacobson (2012)). As it is not expected that these excess cancers can ever be distinguished from the background cancer incidence and mortality, these quoted values will have to be assessed and interpreted with great care.

Table 3b
Selected maximum radionuclide concentrations in rainwater after the accidents of Chernobyl and Fukushima, arranged by distance from the NPPs (activities in $Bq \cdot L^{-1}$ at the time of sampling).

Chernobyl						Fukushima					
Location	Distance to NPP (km)	^{131}I	^{137}Cs	^{90}Sr	$^{239+240}Pu$	Location	Distance to NPP (km)	^{131}I	^{137}Cs	^{90}Sr	$^{239+240}Pu$
						Kashiwa, JP	200	6 072[l]	752[l]	–	–
						Tokyo, JP	250	284[m]	236[m]	–	–
						Tokyo, JP	250	614[l]	27.2[l]	–	–
						Nagoya, JP	400	810[n]	240[n]	–	–
						Osaka, JP	600	0.056[o]	–	–	–
Northern Austria	1100	–	–	320[a]	–	Hiroshima, JP	840	0.44[p]	0.173[p]	–	–
Göteborg, S	1300	3000[b]	950[b]	–	–	Jeju, ROK	1240	2.81[q]	2.02[q]	–	–
Thessaloniki, GR	1300	8400[c]	1700[c]	–	–						
Munich, D	1400	58,000[d]	6500[d]	–	–						
Northern England, UK	2200	200[e]	80[e]	–	–						
Shetlands, UK	2200	1600[f]	290[f]	–	–	Krasonoyarsk, RU	4000	0.62[r]	0.075[r]	–	–
Jerusalem, IL	2200	19[g]	–	–	–	Novosibirsk, RU	4700	0.83[s]	0.092[s]	–	–
Taiwan	7600	1.4[h]	–	–	–	Boise, ID, US	8100	14.43[t]	1.33[t]	–	–
Japan	8100	98[i]	11[i]	0.02[j]	–	San Francisco, US	8100	16[u]	0.5[u]	–	–
						Richmond, CA, US	8100	5.11[v]	0.29[v]	–	–
Arkansas, US	8900	1.7[k]	–	–	0.00019[k]	Vartop, RO	8900	1.69[w]	<0.039[w]	–	–
						Bremen, D	8900	0.068[x]	3.00[x]	–	–
						Vienna, AT	9000	5.2[y]	–	–	–
						Denver, CO, US	9100	1.59[z]	–	–	–
						Thessaloniki, GR	9300	0.7[aa]	–	–	–

– … not detected or not reported
[a] 29 April 1986 (Batarekh and Teherani, 1987).
[b] 10 May 1986 (Mattsson and Vesanen, 1988).
[c] 5–6 May 1986 (Papastefanou et al., 1988; Papastefanou et al., 1989).
[d] 30 April 1986 (Heinzl et al., 1988).
[e] 2 May 1986 (Livens et al., 1992).
[f] 1–6 May 1986 (Cambray et al., 1987).
[g] 3–4 May 1986 (Paul et al., 1987).
[h] May 1986 (Chung and Lo, 1986).
[i] Ibaraki, 6–15 May 1986 (Muramatsu et al., 1987).
[j] Ookuma, 30 April–22 May 1986 (Higuchi et al., 1988).
[k] 10 May 1986 (Kuroda et al., 1989).
[l] 20–22 March 2011 (Hazama and Matsushima, 2013).
[m] 4 April 2011 (Steinhauser et al., 2013a).
[n] 18 April 2011 (Ogata, 2013).
[o] 8 April 2011 (Hazama and Matsushima, 2013).
[p] April 2011 (Hazama and Matsushima, 2013).
[q] 28 March–31 May 2011 (Kim et al., 2012).
[r] 4 April 2011, melted snow (Bolsunovsky and Dementyev, 2011).
[s] 2–3 April 2011, melted snow (Melgunov et al., 2012).
[t] 22 March–27 April 2011, US EPA (Thakur et al., 2012).
[u] 24 March 2011 (Norman et al., 2011).
[v] 23 March–28 April 2011, US EPA (Thakur et al., 2012).
[w] 7–8 April 2011 (Cosma et al., 2012).
[x] 15 April 2011 (Pittauerová et al., 2011).
[y] 26 March 2011 (Steinhauser et al., 2013a).
[z] 4–14 April 2011, US EPA (Thakur et al., 2012).
[aa] 29 March 2011 (Manolopoulou et al., 2011).

Plans for in-depth epidemiological studies of the health consequences of the Fukushima accident were set up quickly after the accident by the Japanese government and local health authorities. The draft plans for the health care of residents and initial implementations have been reviewed by Akiba (2012). One of the big challenges was to establish a cohort of the affected population and of a control group, especially since the doses (already in the first year after the accident) were estimated to be so low that it was difficult to evaluate the risk of cancer and non-cancer diseases (Akiba, 2012; Boice, 2012). Another potential problem is the lack of comprehensive dosimetric radiation surveys (a result of the chaotic conditions and earthquake/tsunami-related destruction of infrastructure), especially of short-lived radionuclides such as ^{131}I (Akiba, 2012). In this case, retrospective dosimetry using long-lived ^{129}I as a tool to reconstruct initial ^{131}I doses may be useful (Michel et al., 2005; Pietrzak-Flis et al., 2003; Straume et al., 2006).

Probably the most striking difference between the Chernobyl and Fukushima nuclear accidents concerns the acute (deterministic) effects on workers at the plant sites. In the course of the Chernobyl accident, clinical evidence indicated that 134 workers suffered from acute radiation syndrome (ARS). Thirty-one persons (according to (Michel and Voigt, 2006)) or 28 (according to (The Chernobyl Forum, 2006; UNSCEAR, 2000)) died from the explosion and ARS in 1986. From 1987 to 2004 another 19 died of various causes. The maximum external doses for emergency workers on the first day of the Chernobyl accident reached up to 16 Gy (UNSCEAR, 2000). In the course of the clean-up, 187,000 workers and liquidators were exposed to a mean effective dose of 170 mSv (Michel and Voigt, 2006; United Nations, 2000).

Data on the exposure of workers at Fukushima NPP are (yet) sparse and/or not validated or reviewed by international organizations such as the IAEA or the WHO and thus may be subject to correction in the future. Of 20,000 workers at the NPP site, 146 received doses >100 mSv. Six workers received doses >250 mSv. Two workers in the control rooms received doses >600 mSv and two other workers received skin doses of 2–3 Sv while standing in highly contaminated water (Ten Hoeve and Jacobson, 2012). In any case, no cases of ARS or radiation sickness have been reported for any of the emergency workers at Fukushima NPP. The excess cancer morbidity among the 20,000 workers was estimated to be 2–12 cases (Ten Hoeve and Jacobson, 2012), which, given the

spontaneous cancer incidence rate, cannot be statistically resolved. In a very recent report, the WHO (World Health Organization, 2013) concluded that the radiation levels in Fukushima prefecture were "well below" the threshold levels for deterministic health effects. Inside the exclusion zone, however, high dose rates (4.8–98 μSv/h) have been reported (Endo et al., 2012), which could potentially lead to high annual cumulative external doses (51–1000 mSv), if not evacuated.

7. Food safety after the accidents

After a nuclear accident and the release of radionuclides into the environment, human health can be affected in several ways, including external exposure, internal exposure due to inhalation of radionuclides and internal exposure due to ingestion of radionuclides with contaminated food (Merz et al., 2013). After the initial phase of a major nuclear accident, when inhalation contributes significantly to the overall exposure, contaminated food is regarded as the most important pathway of radionuclides into the human body — and thus most dose relevant (Hamada and Ogino, 2012). Depending on the biological half-lives of the radionuclides, continuous accumulation in organs can yield high organ doses. This is most critical for ^{131}I that highly accumulates in the thyroid. It has been observed that ^{131}I activity concentrations continuously build-up over the first weeks and that ^{131}I exhibits an apparent half-life in mammals that, in the initial phase after the accident, is even longer than the physical half-life (Steinhauser et al., 2012). A similar build-up has been shown for fish exposed to cesium (Pinder et al., 2009). The different metabolisms of each radionuclide after intake can be described by means of the biological half-life, which is always shorter than the physical half-life as it includes both physical decay and biological excretion. Metabolism studies of radiocesium showed that 10% of the cesium is excreted with a half-life of 2 d, and 90% with a half-life of 110 d (ICRP, 1979; Rühm et al., 1999). In a simplified approach, the biological half-life of radiocesium is often assumed to be in the range of three months or 100 d for adults.

Table 3c
Selected maximum radionuclide concentrations in biological material after the accidents of Chernobyl and Fukushima, arranged by distance from the NPPs (activities in Bq·kg^{-1} and Bq·L^{-1} for milk, respectively, at the time of sampling/measurement). Values are usually activity per fresh mass, unless noted otherwise.

Chernobyl						Fukushima					
Location	Distance to NPP (km)	^{131}I	^{137}Cs	^{90}Sr	$^{239+240}$Pu	Location	Distance to NPP (km)	^{131}I	^{137}Cs	^{90}Sr	$^{239+240}$Pu
Plant material						*Plant material*					
Pripyat/Yanov, UA	4	–	227 900[a]	1,015,000[a]	1457[b]	Fukushima Daichi, JP	0.88	150,000[gg]	1,290,000[hh]	1140[hh]	0.49[ii]
Christinovka, UA	64	–	–	–	1.25[c]	Odaka, JP	16	–	750[hh]	125[hh]	0.17[ii]
Kiev, UA	100	–	12,350[d]	1320[d]		Iitate, JP	35	2000[gg]	10,000[gg]	–	–
						Kyoto, JP	550	3400[ij]	280[ij]	–	–
Styria, AT	1200	640[e]	580[e]	–	–						
Thessaloniki, GR	1300	–	2700[f]	–	–						
Munich, D	1400	–	200[g]	–	–						
Bremen, D	1500	2450[h]	555[h]	–	–						
Northern Italy	1500	164[i]	51[i]	–	–						
Nice, FR	1900	–	3400[j]	–	–						
Harrington, UK	2100	–	5100[k]	–	–						
United Kingdom	2200	200[l]	100[l]	–	–						
Mumbai, IN	5100	9.20[m]	8.00[n]	–	–						
Taiwan	7600	12[o]	–	–	–	Alemeda, CA, US	8100	9.93[kk]	7.04[kk]	–	–
Ibaraki	8100	106[p]	6.8[p]	–	–	Cluj, RO	8700	2.55[ll]	–	–	–
						Bremen, D	8900	3.58[mm]	1.59[mm]	–	–
						Vienna, AT	9000	4.8[nn]	–	–	–
						Agen, FR	10,000	9[oo]	–	–	–
Thyroids						*Thyroids*					
Austria	1100	31,600,000[q]	3100[q]	–	–						
Munich, D	1400	11,400[r]	–	–	–						
Ulm, D	1500	1,200,000[s]	96[s]	–	–						
Ioannina, GR	1500	618[t]	–	–	–						
Birmingham, UK	2200	622 000[s]	656[s]	–	–						
United Kingdom	2200	2,000,000[u]	–	–	–						
Japan	8.000	3000[v]	–	–	–	Cluj, RO	8700	180[ll]	–	–	–
						Northern Austria	9000	1510[pp]	–	–	–
Cow milk						*Cow milk*					
Kiev, UA	100	–	–	0.27[w]	–	Kawamata, JP	46	1512[qq]	18.4[qq]	–	–
						Fukushima prefecture	70	5300[rr]	20[rr]	–	–
Minsk, BY	350	51.6[x]	85.0[x]	–	–	Ibaraki, JP	130	1700[rr]	64[rr]	–	–
Bratislava, SK	1000	758[y]	–	–	–	Kyoto, JP	550	–	0.7[ss]	–	–
Sweden	1150	10[z]	–	–	–						
Northern Albania	1300	3500[aa]	380[aa]	–	–						
Greece	1300	–	130[f]	–	–						
Bremen, D	1500	40,000[bb]	–	–	–						
United Kingdom	2200	400[l]	400[l]	–	–						
Mumbai, IN	5100	2.80[cc]	1.10[cc]	–	–						
New York, US	7500	1.5[dd]	–	–	–	Hilo, HI, US	6500	0.67[tt]	0.70[tt]	–	–
Taiwan	7600	4[o]	–	–	–						
Nagoya, JP	8000	21[ee]	–	–	–	San Francisco, CA, US	8100	2.9[uu]	0.48[uu]	–	–
						Cluj, RO	8700	0.37[ll]	–	–	–
						Bremen, D	8900	0.08[vv]	<0.02[vv]	–	–
Livermore, CA, US	9800	0.37[ff]	–	–	–	Little Rock, AR, US	10,000	0.33[ww]	–	–	–
						Agen, FR	10,000	0.659[xx]	–	–	–

In the first two years after the accident, the direct deposition of radionuclides on the surface of plant leaves is a more relevant pathway for human intake than root uptake. Another initially important pathway for human intake is pasture that is eaten by livestock, causing the transfer of radionuclides to meat and milk. Although rain is known to wash out radionuclides efficiently from the atmosphere and to boost the contamination level of soil, dry deposition causes a higher relative deposition on the plants and hence a higher surface activity on the plant surface (Jacobi, 1988). In this respect, the date of the accident on 11 March favored the Fukushima accident because it happened prior to the main agricultural season. At the time of the Chernobyl accident (26 April), the season had already begun and radionuclides were directly deposited on the surface of agricultural plants and pasture.

For disaster management, the transfer of radionuclides (especially ^{131}I and ^{137}Cs) from dairy cows into milk is of great importance. Measurements (see Table 3c) and models (Zvonova et al., 2009) show that ^{131}I concentrations in milk after the Chernobyl accident, even in more remote areas, easily reached or exceeded activity concentrations of 10000 Bq/L. However, only few radioanalytical data for milk from regions of the former Soviet Union are available, so the maximum activity concentrations in the most affected regions are likely to have been even much higher. Moreover, many people in the contaminated areas lived autarkical on homegrown vegetables and milk from privately owned dairy cows. This massively increased their intake of radionuclides and internal exposure (The Chernobyl Forum, 2006). Consumption of milk contaminated with ^{131}I has been identified as dominating the ingestion dose of the local population after the Chernobyl accident (Diehl, 2003; Pröhl et al., 2002). Due to the lack of radioanalytical equipment and the short half-life of ^{131}I, dose estimations due to radioiodine in milk and other foodstuff often could not be performed in a timely manner and had to be estimated at a later point using the long-lived fission product ^{129}I or the correlation of ^{131}I with ^{137}Cs (which often exhibits large fluctuations) (Michel et al., 2005; Straume et al., 2006). As discussed above, stable iodine prophylaxis after Chernobyl came too late or was not performed at all. Warnings not to consume milk from local livestock were issued days after the accident. Even years after the accident, milk remained the major route for intake of ^{137}Cs and contributed more than 50% to the average intake (Table 4) (Travnikova et al., 2001).

Notes to Table 3c:
– … not detected or not reported
[a]Probably late 1990s, leaves (Victorova et al., 2000).
[b]2006, moss (dry mass) (Bisinger, 2009).
[c]2003, oat (Hippler, 2006).
[d]1990, IAEA Reference material Grass (IAEA-373).
[e]April 1986 (Teherani, 1987).
[f]May 1986–May 1987 (milk); 17 May 1986 (grass) (Papastefanou et al., 1988).
[g]Fall 1986, red currants (Heinzl et al., 1988).
[h]5 May 1986 (Pittauerová et al., 2011).
[i]May 1986 (Beiriger et al., 1988).
[j]17 August 1986, *Parmelia furfuracea*, dry (Barci et al., 1988).
kEnd of May 1986 (Cambray et al., 1987).
[l]Early May 1986, leafy vegetables and milk (Fry et al., 1986).
[m]24 May 1986, grass (Mishra, 1990).
[n]20 May 1986, grass (Mishra, 1990).
[o]May 1986; cow milk and vegetables (Chung and Lo, 1986).
[p]15 May 1986, shungiku (Muramatsu et al., 1987).
[q]22 May 1986, roe deer (Tataruch et al., 1988).
[r]Total activity of a human thyroid 227 Bq measured on 6 May 1986 (Kutschera et al., 1988). Assumed thyroidal mass 20 g (Nishizawa et al., 1988).
[s]21 May 1986, cattle (Ulm); 21 May 1986, sheep (Birmingham) (Van Middlesworth and Loos, 1988).
[t]3 July 1986, sheep (Ioannides et al., 1991).
[u]May 1986, sheep (Sansom, 1989).
[v]20 May 1986, cattle (Van Middlesworth, 1990).
[w]19 June 1986 (Baratta, 2003).
[x]22–29 May 1986 (Baratta, 2003); ^{137}Cs in milk dropped to 2.3 Bq/kg by 24 February 1998 (Baratta, 2003)
[y]16 May 1986 (Koprda, 1990).
[z]Early May 1986 (Devell et al., 1986).
[aa]5–6 May 1986 (Kedhi, 1990).
[bb]1986 (Pittauerová et al., 2011).
[cc]23 May 1986 (Mishra, 1990).
[dd]12–17 May 1986 (Feely et al., 1988)
[ee]19 May 1986 (Nishizawa et al., 1988).
[ff]14 May 1986 (Beiriger et al., 1988).
[gg]10 April 2011, grass (Shozugawa et al., 2012).
[hh]21 December 2011, grass (Steinhauser et al., 2013b).
[ii]21 December 2011, grass (Schneider et al., 2013)
[jj]23 March 2011, leafy vegetable (Banno et al., 2013).
[kk]3–14 April 2011, UCB data (Thakur et al., 2013).
[ll]14 April 2011(grass); 23 April 2011 (sheep thyroid); 12 April 2011 (cow milk) (Cosma et al., 2012).
[mm]13 April 2011 (Pittauerová et al., 2011).
[nn]12 April 2011, grass (Steinhauser et al., 2013a).
[oo]30 March 2011, grass (Parache et al., 2011).
[pp]11 April 2011, European mouflon (Steinhauser et al., 2012).
[qq]MEXT (Thakur et al., 2013).
[rr]19 and 20 March 2011 (Hamada and Ogino, 2012).
[ss](Koizumi et al., 2012).
[tt]4 April 2011, US EPA (Thakur et al., 2012).
[uu]5–6 April 2011, US EPA (Thakur et al., 2012).
[vv]14 April 2011 (Pittauerová et al., 2011).
[ww]30 March 2011, US EPA (Thakur et al., 2013).
[xx]6 April 2011 (Parache et al., 2011).

According to this table, inhabitants of the heavily contaminated Bryansk region had an average daily intake of more than 1500 Bq ^{137}Cs in the first year after the Chernobyl accident (Travnikova et al., 2001). According to Japanese scholars, the daily intake of ^{137}Cs was 0.6 Bq for the inhabitants of the Fukushima prefecture, which is the most contaminated prefecture after the Fukushima nuclear accident (Koizumi et al., 2012) (note that ^{134}Cs intake was slightly higher after Fukushima due to the higher ^{134}Cs/^{137}Cs activity ratio). A 2011 study of inhabitants in that region is in good agreement with these results as the authors found an average daily intake of 2.5 Bq ^{137}Cs (Harada et al., 2012).

The contamination with ^{90}Sr and α-emitting actinides outside the 30 km radius zone of Chernobyl was low (though measurable) and did not contribute significantly to the internal exposure (Cooper et al., 1992; Diehl, 2003). The short(er)-lived radionuclides ^{131}I and ^{134}Cs decayed within the first months or the first decade after the accident, respectively. Later, longer-lived ^{137}Cs dominated the contamination of foodstuff (Table 4). Wild mushrooms accumulate radiocesium to a great extent and lead to remarkable activity concentrations beyond 100,000 Bq/kg (Cooper et al., 1992), which poses additional risk to the inhabitants (Travnikova et al., 2001), who have a long tradition in collecting forest mushrooms.

Although it is difficult to come to ultimate conclusions for every individual of the affected Japanese population in affected areas, recent literature (Hayano et al., 2013; Normile, 2013) indicates that the efforts to keep above-limit contaminated food off the market after the Fukushima accident were successful. Japanese authorities ordered the analysis of thousands of food samples immediately after the accident as well as an efficient monitoring network (Hamada and Ogino, 2012; Hamada et al., 2012; Merz et al., 2013). In order to increase the throughput of samples, universities and research institutes participated in the monitoring campaigns. The measurements were further accelerated by assuming intrinsic co-existence of gamma-emitting radiocesium (which can be determined quickly) with other radionuclides such as ^{90}Sr (a pure beta emitter) and α-emitting actinides (primarily $^{239+240}$Pu) (Merz et al., 2013). For both ^{90}Sr and $^{239+240}$Pu, this approach was found to be correct (Schneider et al., 2013; Steinhauser et al., 2013b). Since the emissions of ^{90}Sr and plutonium from Fukushima Daiichi were found to be negligible, the new regulatory limits did not include these radionuclides. The regulatory limits in place for food immediately after the accident were more restrictive than European limits, and since then have been further reduced (Merz et al., 2013).

Hayano et al. (2013) investigated the internal ^{137}Cs exposure of almost 33000 Fukushima residents using whole-body counters. They found very low radiocesium concentrations in the bodies of Miharu-town school children (50 km west of Fukushima NPP) under the age of 15: in winter 2011, 54 of 1494 tested children (coverage of 94.3%) had detectable radiocesium levels, reaching up to 1400 Bq/body. The activity in the vast majority of the children did not exceed the detection limit of 300 Bq/body. In fall 2012, none of 1383 tested children (coverage 95.0%) exceeded the detection limit. Four senior residents, however, showed ^{137}Cs levels of up to 12240 Bq/body, causing a maximum committed effective dose of 1.06 mSv. After the Chernobyl accident such levels of contamination were frequently observed throughout the population. Similar to the Chernobyl experience, the source of the high levels in the four adults from Fukushima prefecture was the consumption of wild mushrooms, wild boar meat and freshwater fish they had obtained themselves, bypassing the mandatory food monitoring in markets (Normile, 2013).

8. Mutual characteristics of both accidents

Both accidents at Chernobyl and Fukushima involved civil power reactors and caused the release of mainly volatile radionuclides. Beyond these most obvious technological common grounds, the accidents have some interesting mutual socioeconomic characteristics. Both accidents were comparable in their public perception, causing widespread loss of trust in nuclear technology, perhaps beyond a scale some nuclear scientists would have accepted as justified. Both accidents also had a strong political dimension: Chernobyl, for example, ultimately resulted in a cessation of the Austrian nuclear power program (Khan et al., 2010); Fukushima initiated Germany's abandoning of nuclear energy (Gross, 2011). Both accidents initiated public mistrust in governmental actions in the aftermath of the disasters (Normile, 2013).

Both accidents also had a health dimension beyond the radiological risk. The Chernobyl Forum reported, "Life expectancy has declined precipitously, particularly for men, and in the Russian Federation stood at an average of 65 in 2003 (just 59 years for men). The main causes of death in the Chernobyl-affected region are the same as those nationwide — cardiovascular diseases, injuries, and poisonings — rather than any radiation-related illnesses. The most pressing health concerns for the affected areas thus lie in poor diet and lifestyle factors such as alcohol and tobacco use, as well as poverty and limited access to health care" (The Chernobyl Forum, 2006). Nature pointedly concluded from this paragraph that poverty, anxiety, depression, and stress "posed bigger threat than radiation" (Valeska, 2005), which initiated a public controversy (Mousseau et al., 2005). However, other authors also reported on long-term psychological effects caused by the accident (Adams et al., 2011; Bromet et al., 2011a; Bromet, 2012; UNSCEAR, 2008). Bromet et al. (2011b) state that "mental health effects were the most significant public health consequence of the accident". Similarly, Michel and Voigt (2006) conclude that the socio-ethical and political consequences of Chernobyl outweighed the effects of radiation.

For Fukushima, similar effects are possible and in part already visible (Von Hippel, 2011). Brumfiel (2013) discusses case studies that led him to conclude that "Japan kept people safe from the physical effects of radiation — but not from the psychological impacts." Such impacts include effects such as anxiety, fear, depression but also unexplained physical symptoms. They may have been triggered or enhanced by widespread mistrust of the Japanese Government (Brumfiel and Fuyuno, 2012; Ten Hoeve and Jacobson, 2012). Psychological effects and posttraumatic stress response, however, were not restricted to the evacuees and the normal population alone, they also occurred in workers at the Fukushima NPP (Shigemura et al., 2012a, 2012b). Fear of radiation also caused many under 40 to move out of the affected areas in Japan, which not only caused a significant depopulation but also a rapid aging of the remaining population, as shown for Minamisoma City (Ishikawa et al., 2012). Among the evacuees, three-fourths of the affected families still live in separation, because one parent works in a more contaminated area, whereas the other parent and the children moved elsewhere.

While writing this review and searching for literature, we also encountered one interesting and unexpected similarity in the scientific evaluation of both accidents: a significant portion of literature is written in the respective mother-language of the affected countries (Russian and Japanese, respectively), which makes the dissemination of scientific results within the radiological and radioecological community difficult. The database SciFinder® currently lists 10,974 results for the search term "Chernobyl" (1986-2013). 20% (2193) of these papers are written in Russian. For the search term "Fukushima" (2011–2013), currently 1738 results are listed, of which 22% (385) are written in Japanese. In terms of communicating the lessons to be learned by the community, we encourage researchers from Japan and from the former Soviet Union, to seek help publishing their results in English.

9. Conclusion

In summary, the environmental and radiological consequences caused by the Chernobyl nuclear accident clearly exceeded those of the Fukushima accident in almost every respect. In 2006, IAEA classified the Chernobyl nuclear accident as the "foremost nuclear catastrophe in

Table 3d
Selected maximum radionuclide concentrations in soil after the accidents of Chernobyl and Fukushima, arranged by distance from the NPPs (activities in Bq·kg^{-1} at the time of sampling/measurement).

Chernobyl						Fukushima					
Location	Distance to NPP (km)	^{131}I	^{137}Cs	^{90}Sr	$^{239+240}$Pu	Location	Distance to NPP (km)	^{131}I	^{137}Cs	^{90}Sr	$^{239+240}$Pu
Pripyat/Yanov, UA	4	–	1,239,000[a]	420,000[b]	–	Fukushima Daiichi, JP	0.88	49,000[l]	1,790,000[m]	1070[m]	<2.211[n]
Novo-Shepelichi, UA	5	–	–	–	314[c]	Fukushima Daiichi, JP	4.3	10,000[l]	2,740,000[m]	232[m]	<0.5[n]
Christogalovka, UA	5	–	74,000[d]	36,000[d]	–	J village	20	20,030[o]	11,480[o]	–	–
						Kawamata, JP	40	2810[p]	9380[p]	–	–
Christinovka/Polesskoye, UA	64	–	15,000[d]	130[d]	17.8[e]	Fukushima prefecture	60	–	325[q]	–	–
Kupetsch, UA	100	–	3460[f]	44[f]	1.141[e]	Sendai, JP	95	–	5000[m]	<3[m]	–
Minsk, BY	350	–	257.4[g]	–	–	Kashiwa, JP	195	–	421,000[m]	35[m]	<0.53[n]
Vöcklabruck, AT	1250	–	506[h]	–	–						
Athens, GR	1600	–	22,000[i]	–	–						
Mercantour Mass., FR	1900	–	1500[j]	–	–						
Mumbai, IN	5100	1.46[k]	9.5[k]	–	–	Mekong River Delta, VN	4700	–	35[r]	–	–
						Bremen, D	8900	0.068[s]	3.00[s]	–	–
						Vienna, AT	9000	<0.4[t]	21.4[t]	–	–

– … not detected or not reported.
[a] 1992 (Konoplyova et al., 1993).
[b] (Victorova et al., 2000).
[c] 1996 (Hippler, 2006).
[d] (Malek et al., 2001).
[e] 2003 (Hippler, 2006).
[f] Summer 1996 (Filss et al., 1998).
[g] 25 February 1992 (Baratta, 2003).
[h] April 1992 (Steinhauser et al., 2013a).
[i] May–November 1986 (Simopoulos, 1989).
[j] June 2002 (Rezzoug et al., 2006).
[k] 26 May 1986 (Mishra, 1990).
[l] 10 April 2011 (Shozugawa et al., 2012).
[m] 21 December 2011 (Steinhauser et al., 2013b).
[n] 21 December 2011 (Schneider et al., 2013).
[o] 20 April, 2011 (Tagami et al., 2011).
[p] 28 April 2011 (Kato et al., 2012).
[q] (Oshita et al., 2013).
[r] 29–31 October 2011, Coastal sediment of the Mekong River Delta (Kanai et al., 2013).
[s] 4 April 2011 (Pittauerová et al., 2011).
[t] 12 April 2011, please note that no ^{134}Cs was detected (old Chernobyl, fallout ^{137}Cs) (Steinhauser et al., 2013a).

Table 4
Cesium-137 contamination in selected food products collected after the Chernobyl accident in Veprin (Bryansk region, Russian Federation) from 1994 to 1998 and average daily intake (Bq per day) of ^{137}Cs from various foods by the inhabitants of Veprin. Data taken from Travnikova et al. (2001).

Product	Average ^{137}Cs conc. (Bq·kg^{-1})	Average daily intake							
		1987		1990		1993		1996	
		Bq/d	%	Bq/d	%	Bq/d	%	Bq/d	%
Cow milk	315	1109	70.2	175	57	240	62.8	166	54.3
Meat	680 (beef) 194 (pork)	190	12.0	33	10.7	36	9.4	35	11.5
Eggs	N.R.	13	0.8	1.2	0.4	2.3	0.6	0.9	0.3
Potatoes	45	122	7.7	24	7.8	21	5.5	21	6.9
Other vegetables	29 (carrots) 24 (beet) 35 (cabbage) 21 (tomatoes) 67 (haricot beans) 126 (sorrel)	44	2.8	5.2	1.7	6.1	1.6	5.8	1.9
Fruits	13 (apples) 3720 (wild bilberry) 190 (raspberry)	6.3	0.4	0.1	0.03	1.1	0.3	2.1	0.7
Wild mushrooms	2550 (chanterelle) 14,500 (boletus luteus) 8980 (russula)	66	4.2	66	21.5	66	17.3	66	21.6
Fish	110 (roach) 240 (pike) 252 (river crucian) 730 (lake crucian) 141 (perch) 96 (tench)	28	1.8	2.5	0.8	9.5	2.5	8.9	2.9
Total		1580	100	307	100	382	100	306	100

N.R. Not reported.

human history" (The Chernobyl Forum, 2006); a perception that still remains valid even after the Fukushima nuclear accident. Although the Fukushima accident involved four reactors and the Chernobyl accident "only" one, we estimate the total release of radionuclides from Fukushima NPP (approximately 520 PBq) to be approximately one order of magnitude lower than the release from Chernobyl (approximately 5300 PBq) (UNSCEAR, 2000). Consequently, the contamination levels in all relevant biota (air, rainwater, soil, vegetation, cow milk and animal thyroids) were in the vast majority of cases significantly higher after the Chernobyl accident than after Fukushima, when the distance to the reactors is taken into account. Also, the size of the evacuation areas and the highly contaminated areas are much larger around Chernobyl compared to Fukushima. Most importantly, the projected number of radiation-induced casualties or cancer morbidities amongst the Fukushima emergency workers and the population is too low for direct observation in epidemiological studies due to the lack of statistical significance of the expected small increase in the number of cancer cases. This is in stark contrast to Chernobyl, where several thousand (thyroid) cancer cases can be attributed to radiation effects among children, adolescents and since recently an increase in leukemia cases among the "liquidators". Food safety campaigns after the Fukushima accident have proven their efficiency keeping most of the contaminated food off the market. In contrast, inhabitants of affected zones around Chernobyl continued consuming local products (especially milk) until a great deal of harm had been done before warnings were issued. Due to lower radiation levels after the Fukushima accident, no cases of acute radiation syndrome were reported among workers compared to the 134 confirmed cases of ARS after Chernobyl. Fukushima did not cause any casualties due to acute radiation, whereas the very high levels of radiation at Chernobyl caused the death of approximately 50 persons. Socioeconomic effects, such as widespread mistrust in nuclear technology as well as non-radiation-induced diseases such as anxiety, depressions, or posttraumatic stress are likely to be the most significant effects from both accidents.

Acknowledgments

This publication was supported by Grant Number T42OH009229-07 from CDC NIOSH Mountain and Plains Education and Research Center. Its contents are solely the responsibility of the authors and do not necessarily represent the official views of the CDC NIOSH and MAP ERC.

Appendix A. Supplementary data

Supplementary data associated with this article can be found in the online version, at http://dx.doi.org/10.1016/j.scitotenv.2013.10.029. These data include Google maps of the most important areas described in this article.

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